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Review Paper

First, do no harm: A systematic review of deforestation spillovers from protected areas

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ABSTRACT

Land-use restrictions in protected areas (PAs) might have unintended spillover effects on non-target, neighboring areas. In the case of leakage, land-use change that would have occurred in the PA is displaced to an unprotected area where it would not have otherwise occurred. The resultant 'leakages' can offset benefits achieved inside PAs, confound impact assessments, and exacerbate the problems of opportunistic protection of lands. Conversely, in the case of 'blockage', the unprotected surroundings experience less land-use change than would have otherwise occurred due to a positive spillover effect from nearby protected areas. Little is known about the magnitude, ubiquity, and predictability of spillovers that have already occurred in the global PA network. Here we systematically review the literature and collate the existing evidence to quantify deforestation spillovers. We calculated deforestation rates within 3,398 PAs, most of which were found in tropical and subtropical moist forests, and in their unprotected adjacent surroundings and compared these rates to a baseline derived from the wider landscape. Of the 2,575 PAs that effectively restricted deforestation rates within their bounds relative to the baseline, 11.8% showed leakage and 54.8% exhibited blockage. Deforestation rates in the remaining 33.4% were indistinguishable from their respective baselines. Linear modelling of the correlates of leakage and blockage showed that PA-specific characteristics like size and IUCN category were uninformative, whereas national-scale socioeconomic factors like population density and GDP were useful predictors. Although spillovers from land-use restrictions are ultimately driven by socioeconomic factors, their ecological consequences are such that PA assessments should routinely and explicitly account for displaced impacts to the unprotected surroundings.
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1. Introduction

The expanding global protected area (PA) network acts as a primary bulwark against encroaching anthropogenic threats to biodiversity, and its importance is codified into international law: Aichi Target 11 of the 2010 Convention on Biological Diversity (CBD; http://www.cbd.int/sp/targets) called for the protection of 17% of the planet's terrestrial surface by 2020, and this goal is likely to be achieved (Tittensor et al., 2014). The well-developed field of systematic conservation planning (Margules and Pressey, 2000) provides a framework for implementing and managing this significant investment by

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governments to restrict or ban human land use for the benefit of biodiversity. Evaluating PAs' effectiveness is difficult, because their establishment can be motivated by multiple agendas (Naughton-Treves et al., 2005), but their ability to prevent forest conversion can easily be assessed empirically; as such, this is a commonly used metric for ensuring that these conservation efforts provide tangible benefits. To date, published assessments have demonstrated repeatedly that PAs experience significantly lower rates of forest clearing in comparison to their unprotected surroundings (Nagendra, 2008; Naughton-Treves et al., 2005). Yet, such assessments of PA effectiveness rarely acknowledge the possibility of spillover effects, whereby protection in one area impacts the nontarget neighbouring areas, with potentially negative implications for biodiversity (Bode et al., 2015; Renwick et al., 2015).

The possible impact of terrestrial PAs on their unprotected adjacent surroundings[1] (UAS) can be divided into three categories: leakage, blockage, or no effect. Leakage, the problematic type of spillover, occurs when land use restricted in one area is displaced to an unprotected area where it would not otherwise have occurred, diminishing or even entirely offsetting habitat conservation achieved within a PA (Ewers and Rodrigues, 2008). Leakages are well understood in the context of carbon offsetting (Murray et al., 2004). Conversely, blockage incurs an unintended beneficial outcome; for example, in the context of marine protected reserves, where the primary nontarget effect observed is increased fisheries productivity near reserve bounds (Harmelin-Vivien et al., 2008), spillover effects are generally considered to be a positive side effect. In the terrestrial context, blockage occurs when land use is restricted in the target area whilst also occurring at lower rates than would be expected in the corresponding UAS (Herrera Garcia, 2015). Blockage can manifest, in the context of a forested landscape, as a "halo" of limited clearing directly surrounding a PA in a more heavily cleared broader landscape; such halos may be due to protection in the region discouraging investment in transportation infrastructure and industry (Herrera Garcia, 2015). Although *de facto* protection associated with a blockage effect is ostensibly beneficial for habitat conservation, it is unlikely to be permanent. Where no spillover occurs, rates of land use in the UAS are simply concordant with those that would be expected without nearby protection (e.g. Andam et al., 2008).

Land-use spillovers from terrestrial PAs are rarely measured and reported directly (Miteva et al., 2012; Pfaff and Robalino, 2017) although spillovers are both quantifiable (Ewers and Rodrigues, 2008) and have important implications for biodiversity (Bode et al., 2015; Renwick et al., 2015). If PA impact assessments do not account for leakage where it is occurring, the difference between the deforestation rate in the PA and the rate in the surroundings can be inflated as a consequence, and thus the assessment will overstate the PA's land-use restriction success and will fail to acknowledge the negative impact that results in the UAS (Ewers and Rodrigues, 2008). Furthermore, leakage could involve displacement of human land use from areas of low biodiversity value to areas containing more important habitats of more threatened species (Renwick et al., 2015). This perverse effect is plausible, given that PAs have historically been established in places unlikely to be developed for human land use rather than in top priority areas for biodiversity conservation (Joppa and Pfaff, 2009; Venter et al., 2018).

An understanding of the direction, magnitude, ubiquity, and predictability of past spillovers is critical for informing the future placement and maintenance of the PAs, if they are to reach the CBD's goals for landscape protection and biodiversity conservation. Here we systematically review the evidence for deforestation spillovers, especially leakages, from land-use restrictions associated with PAs to determine the extent to which PA establishment has done "more good than harm" (Pullin and Knight, 2009). We sought to answer the following questions:

a Of PAs that successfully restricted deforestation within their bounds, how many demonstrated a positive (blockage) versus negative impact (leakage) on their unprotected adjacent surroundings?
b Is the magnitude of spillover, whether leakage or blockage, related to characteristics of the PA (e.g., size or IUCN category) or to national-scale socioeconomic factors related to deforestation pressure (e.g., population density and the extracted forest products' economic value)?

The "unholy trinity" of challenges facing efforts to reduce deforestation is to simultaneously achieve additionality, permanence, and avoidance of leakage (Van Oosterzee et al., 2012). Failure to account for leakage where it is likely to occur would violate conservation's version of medicine's principle of "first, do no harm".

2. Methods

2.1. Search strategy and evidence screening

An *a priori* systematic review protocol was devised following guidelines specific to conservation and environmental management (Pullin and Stewart, 2006). Four databases were searched between January and March 2018: Web of Science, Scopus, Google Scholar, and the ProQuest thesis and dissertation database (Table 1). No topical or geographic restrictions were imposed.

Unique records identified by the database searches were screened manually by title for relevance to conservation science or policy. Those records relevant to conservation were then screened using the abstract/summary for any mention of spillover

[1] The unprotected adjacent surroundings are hereafter referred to by the acronym UAS; this acronym is useful for specifying the unprotected area being assessed for local spillover effects, which is comprised of all area falling within a given radial distance out from a PA boundary.

Table 1
Search terms for retrieval of studies. The Boolean operator 'OR' linked terms within categories while 'AND' linked terms between categories; asterisks indicate wildcards and return any use of the root word; quotation marks prevent the search from splitting a phrase into its word components.

Intervention	Outcome	Metric
"protected area*"	leakage	"land use*"
OR ...	OR ...	OR ...
"PA*"	displacement	"deforestation*"
"national park*"	spillover	clearing
"conservation*"	"displaced effort"	"human impact*"
"habitat*"	"protected area isolation"	"anthropogenic pressure*"
"biodiversity*"	"buffer zone*"	"anthropogenic threat*"
"indigenous land*"	"buffer area*"	

(or alternate terminology; Table 1) caused by a conservation or environmental policy or intervention. Finally, full-text screening was used on the filtered list to identify records that provided a sufficient quantitative measure of PA impacts to enable data extraction (or data request). To ensure the comprehensiveness of the literature search, we also screened the bibliographies of the resultant records. An independent reviewer affirmed inter-rater screening reliability.

2.2. Data extraction

According to Ewers and Rodrigues (2008), spillover that occurs as a consequence of PA establishment can be quantified given a measure of land use (e.g., deforestation rate) over the same period in three separate areas of the landscape: a) within the PA, b) in the unprotected adjacent surroundings (UAS) of the PA, and c) in representative parcels from the wider landscape, from which a 'baseline' deforestation rate can be derived (Fig. 2a). If the area, or deforestation rate was not reported in the main text or supplementary information, these data were requested from the corresponding author. A few studies reported alternative proxy measures of land use, like human footprint (Tapia-Armijos et al., 2017), but these (Beresford et al., 2013; Borgström et al., 2012; Joppa and Pfaff, 2011; Paiva et al., 2015; Seiferling et al., 2012) were excluded during the full-text screening (Fig. 1). Although both forest degradation and forest clearing indicate human land use, only deforestation values were extracted because it was the most consistently reported metric. Data were extracted regardless of the number of years between PA establishment and the deforestation monitoring period due to data availability, but ideally, deforestation monitoring would begin prior to, or immediately following, PA establishment. We converted extracted deforestation data to average annual percent of area deforested in order to standardize across PAs of varying size and duration of monitoring.

2.3. PAs' target and non-target impact assessment

Some studies, including Spacklen et al.'s, did not quantify spillovers directly, because leakage detection was not the chief objective of the study; however, they did report relevant data from which we calculated a PA's unintended, non-target deforestation outcomes. We first derived a counterfactual baseline rate of deforestation, i.e. an estimate of what would have happened without legal protection or nearby legal protection, based on the assumption that the average deforestation rate reported in the sample of the broader landscape (Fig. 2) was representative of the likelihood of deforestation in the PA and UAS in the same time period. Given the paucity of studies, we extracted or calculated the spatial baseline in this way regardless of how the landscape sample was chosen, so long as the sample was separate from the UAS zone. However, these samples would ideally be selected with matching methods in order to improve the comparability of this "control" area and the protected "treatment" area (Andam et al., 2008; Mas, 2005).

We then confirmed beneficial PA impact within the target area by identifying those PAs within which the deforestation rate was lower than for the wider-landscape samples. Those for which this was not true may be considered "paper parks," PAs for which gazettal is not concomitant with staffing, management, or enforcement (Di Minin and Toivonen, 2015). Since in these PAs protected designation did not inhibit deforestation, we infer that displacement of deforestation pressure to the surroundings is unlikely.

Following removal of the possible paper parks (Fig. 4), PAs' non-target impact was calculated as the difference of the average deforestation rate observed in the UAS and the baseline rate for the same period. These nontarget impact values, i.e. the spillover magnitudes, were categorised as blockage when a smaller deforestation rate was observed in the UAS than was expected; as leakage when a greater deforestation rate was observed than was expected; or as negligible spillover when the difference in the observed and expected average annual percent deforestation rates was smaller than 0.01.

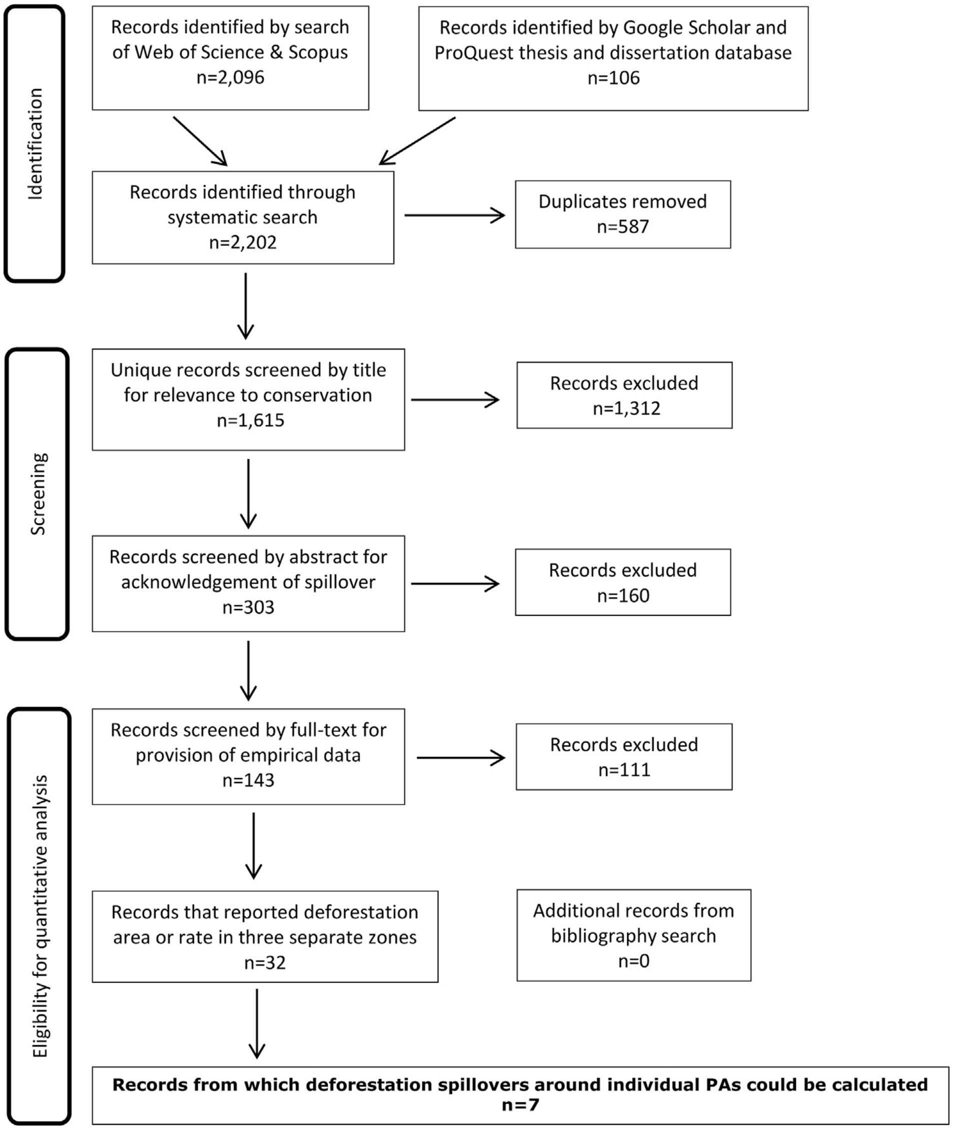

Fig. 1. Screening protocol and numbers of papers included at each step in the systematic review process. Diagram modified from Moher et al. (2009).

2.4. Statistical analysis

We used generalized linear modelling to evaluate fifteen potential predictors of spillover magnitude. Those specific to individual PAs (area, age at the time of deforestation monitoring, and IUCN category) were obtained directly from the papers' main text, from their online supplementary information, or from the World Database on Protected Areas (WDPA). National-scale socioeconomic and political factors ($n = 11$) relevant to deforestation pressure (Table 2) were collected for each country containing one or more of the PAs for which a spillover value was calculated. Where possible, the average over the years 2000–2012 was calculated to reflect the period over which deforestation was monitored for most of the PAs. The continent in which a PA was located was also evaluated as a predictor of spillover magnitude. Three ostensibly important study-specific factors—deforestation monitoring duration in years, UAS definition, and wider landscape sampling method—could not be investigated due to the small number of studies.

Leakage and blockage magnitudes (including those classified as negligible in Fig. 4) were analysed separately as logit-transformed response variables. Two approaches were used to remove observations with missing data, as all fifteen potential predictor variables were not available for all 2,575 PAs. The first was a subset of observations with complete cases, whereby every predictor was available for the observation (no missing values). A second, larger data set was created by removing two factors, PA age and IUCN category (for which many case studies lacked data), and then deleting individual observations that were missing other predictors. Using these two data frames, we fitted generalised linear models to predict:

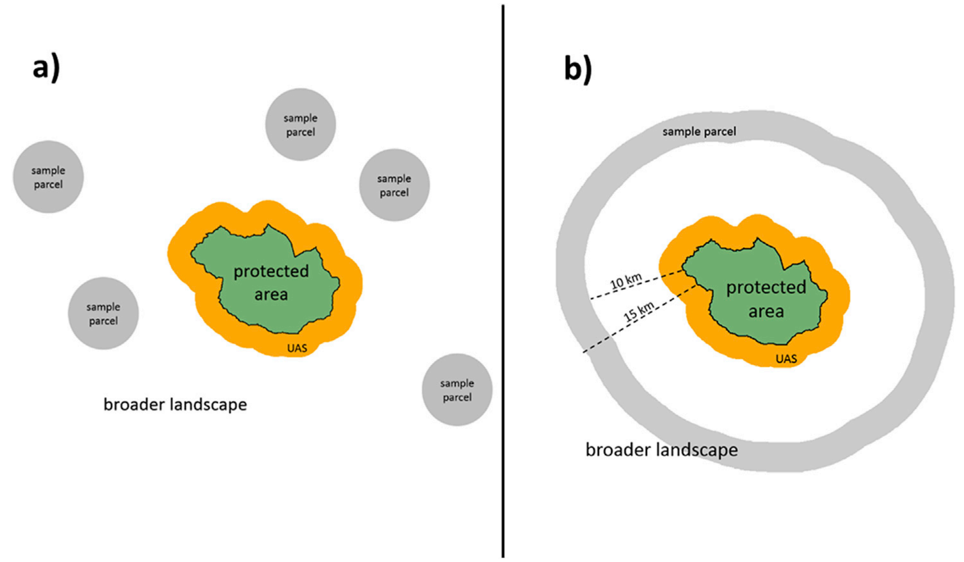

Fig. 2. Map of a hypothetical landscape in which spillover from a PA to the unprotected adjacent surroundings (UAS) can be calculated. The sample of the broader landscape used to derive the baseline deforestation rate could be obtained **a)** as proposed by Ewers and Rodrigues (2008), or **b)** as provided by Spracklen et al. (2015).

(i) leakage, and (ii) blockage outcomes. All subsets of predictor variables were tested using the leaps library (Lumley, 2017) in R (R Core Team, 2018), with k-fold (k = 10) cross-validation (James et al., 2013) used to select the combination that build the model with the best 'hold-out' predictive performance, thereby identifying the most important variables related to spillover magnitude.

3. Results

3.1. Evidence from the systematic search

The literature search yielded 1,615 unique papers, 143 of which addressed spillovers in the context of conservation interventions (Fig. 1); these interventions ranged from payments for ecosystem services in Mexico (Alix-Garcia et al., 2012) and zero-deforestation cattle agreements in Brazil (Alix-Garcia and Gibbs, 2017) to the establishment of PA networks in various countries—the latter being the focus of this review. We identified studies that directly addressed the possibility of land-use leakage from PAs, but which deemed quantitative assessment beyond their scope (Barber et al., 2014; Nepstad et al., 2006; Pfeifer et al., 2012). After excluding records that did not report, nor provide sufficient data to calculate deforestation spillovers around PAs, only seven studies remained (Fig. 1). These (Ament and Cumming, 2016; Apan et al., 2017; Dewi et al., 2013; Phua et al., 2008; Sanchez-Azofeifa et al., 2003, Sánchez-Azofeifa et al., 2009; Spracklen et al., 2015) contained relevant data for 3,404 PAs (Appendix S2), of which 3,374 were reported in Spracklen et al.'s (2015) global-scale study of moist subtropical and tropical forest PAs. The literature search yielded no case studies from PAs in Europe and relatively few (n = 66; 1.9%) from Australia or Oceania (Fig. 3).

Although all seven relevant studies reported deforestation rates within the PA(s), within the UAS, and in a sample of the wider landscape, the definition of the latter two areas was not standardized across studies. The UAS was defined as either 1, 2, 5, or 10-km of radial distance out from PA bounds, depending on the authors' idiosyncratic choice, with the most frequent being 1 km, as chosen by Spracklen et al. (2015). The area chosen to represent the broader landscape likewise varied across studies, with the most common choice being a band spanning 10–15 km beyond the given PA's boundary (Fig. 2b).

3.2. Target and nontarget impacts on deforestation rates

After removal of six PAs due to inadequate spatial data and 823 PAs that did not reduce deforestation rates within their boundaries relative to the rates in the wider landscape samples (Appendix S3), spillover effects to the UAS were quantified for 2,575 PAs. Blockage outcomes far outnumbered leakage outcomes. The deforestation rates in 2,004 (77.8%) of these 2,575 PAs were associated with lower than expected deforestation rates in the UAS, whereas for 571 (22.2%) they were higher than the associated baselines. After removing 860 (33.4%) PAs where the protective effect of the UAS was close to zero, we identified 304 PAs (11.8%) with evidence of leakage and 1,411 (54.8%) PAs showing blockage (Fig. 4). Leakages were least frequent for PAs irrespective of IUCN category, land-use restriction types, sizes, ages or continents (Fig. 5a–e).

Table 2
National scale meta-data factors, their sources, and rationales for their inclusion.

Potential national-scale predictor	Source	Rationale
Population density (people per sq. km of land area) Population growth (annual %) Rural population growth (annual %) GDP growth (annual %) GDP (USD) Ratio of percent of land area under agriculture to percent land area under forest	Arithmetic mean of data reported by the World Bank for the years 2000–2012 (inclusive)	Human development pressures include economic, agricultural and population growth (Waldron et al., 2017).
Total value (USD) of all forest products removed in 2005 Value (USD) of the forest products removed per hectare in 2005	The Food and Agriculture Organisation of the United Nations	These factors are specific indicators of timber and other forest resource extraction pressure.
Carbon dioxide emissions (kilotons)	Arithmetic mean of data reported by the Carbon Dioxide Information Analysis Centre for the years 2000–2010	Changes in land use, mostly deforestation, have been a significant component of anthropogenic carbon emissions from 1960 to 2015 (Le Quéré et al., 2016).
Percentage of total energy supply from solid biofuels in 2010	Arithmetic mean of data reported by the International Energy Agency 2000, 2002, 2005, and 2010	Biofuel plantations can cause indirect land-use changes resulting in deforestation (Lapola et al., 2010) perhaps especially if a country derives a large proportion of its energy supply from biofuels.
Annual conservation spending	*Nature* publication by Waldron et al. (2017)	This factor was included as an indicator of conservation effort, a counterpart to the indicators of human development pressure.

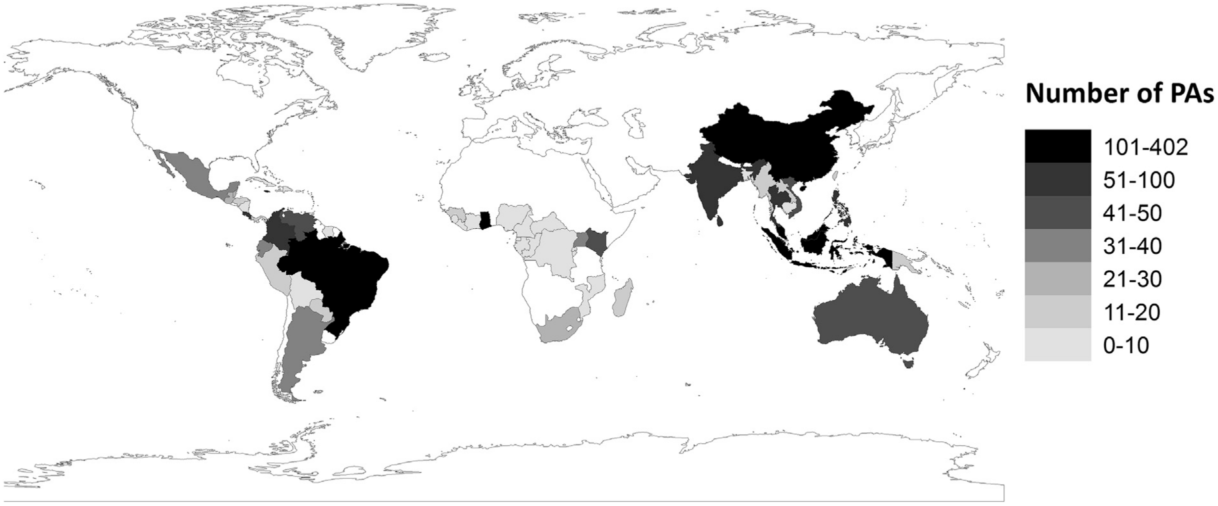

Fig. 3. Map of the countries where deforestation rates within and around PAs, with which spillover could be calculated, were reported in the refereed literature. The shading indicates the number of PAs per country.

3.3. *Drivers of spillover*

We identified 165 leakage observations for which all the chosen predictor variables were known (complete cases). The best-selected model had a structural goodness-of-fit of 14.4 %DE (percent deviance explained) and included six of the potential predictors (Table 3; 'Leakage - Complete cases of 15 variables'). After removal of two data-poor predictors (PA age at time of deforestation monitoring and land-use restriction type), the best model (from 395 observations), with 13.4 %DE, included just two coefficients (Table 3; 'Leakage - Complete cases of 13 variables'). Leakage magnitudes were positively correlated with population growth rate, the ratio of the percent of the country under agricultural cover to the percent under forest cover, and the total and per-hectare values of a country's forest products removed annually. Leakages were negatively correlated with national population density, rural population growth rate, and GDP annual growth rate.

For the 549 observations of blockage where all predictors were available, the best-selected model had 17.3 %DE and included six predictors (Table 3; 'Blockage – Complete cases with 15 variables'). The removal of the same two data-poor variables as for leakage enabled modelling of 1,443 blockage observations, with a 15.9 %DE for the best model and ten predictors (Table 3; 'Blockage- Complete cases with 13 variables'). Blockage was positively correlated with population growth

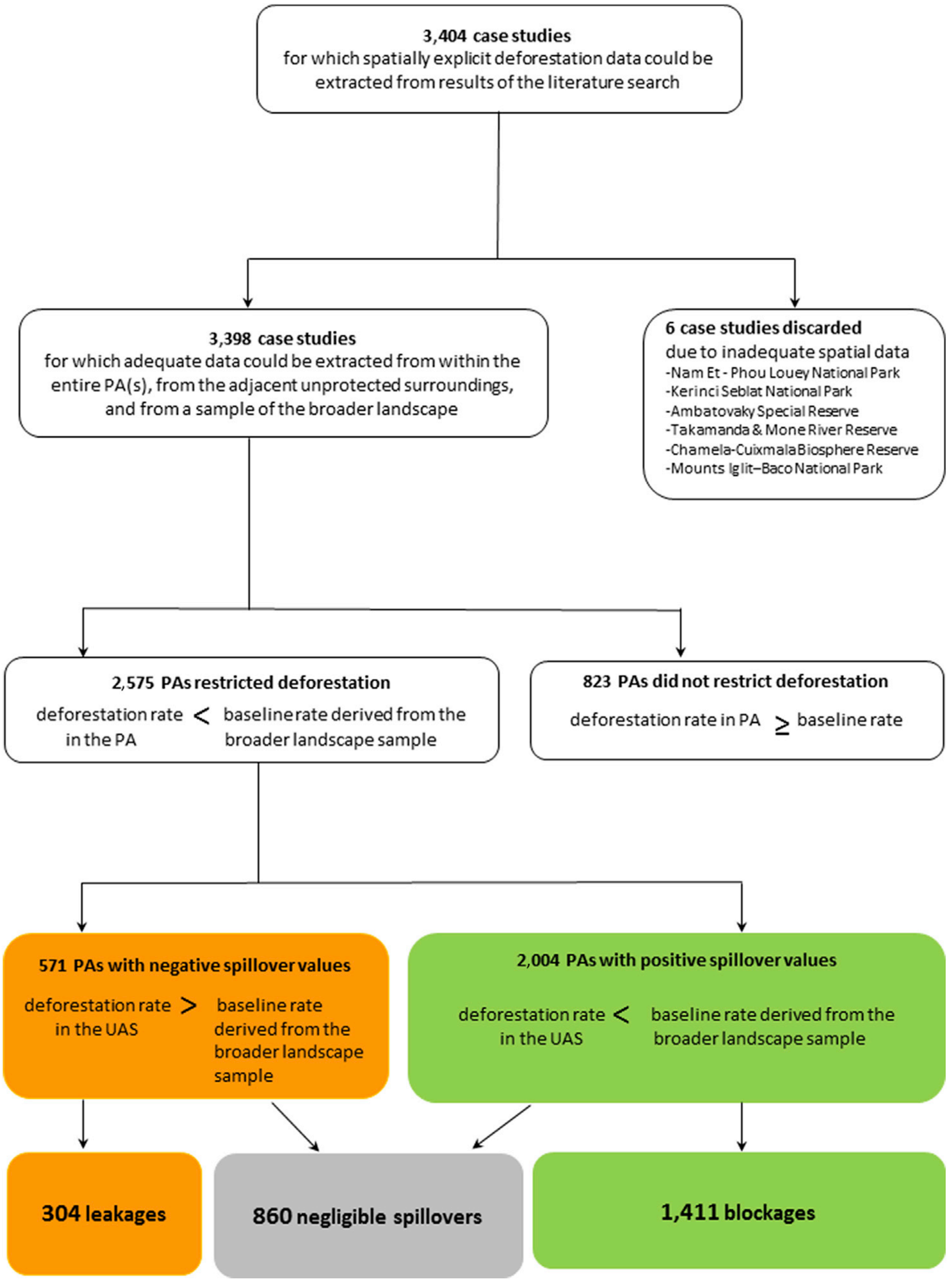

Fig. 4. Flowchart of the stepwise results of assessing PA impact on deforestation rates, indicating the number of PAs that were extracted from the evidence search, that passed data cleaning, that had positive versus negative impact within the target area, that had negative versus positive impact in nontarget area, and that were assigned each spillover outcome type (leakage, negligible spillover, or blockage). Blockage outcomes outnumbered leakage outcomes.

rate, GDP, the ratio of the percent of the country under agricultural cover to percent under forest cover, the per-hectare value of forest products removed in the country annually, and whether a PA was located in Asia or in Australia and Oceania. They were negatively correlated with national population density, rural population growth rate, GDP growth rate, annual carbon dioxide emissions, and a country's annual conservation spending.

The only predictor common to all four best-selected models was the associated country's population density. All showed >13 %DE of their respective data sets (see Appendix S7), and all included only national-scale factors. None of the four best models included coefficients for PA area, PA age, land-use restriction type (derived from IUCN category), the percentage of a country's energy supply from solid biofuels, or PA location within Africa, North America, or South America.

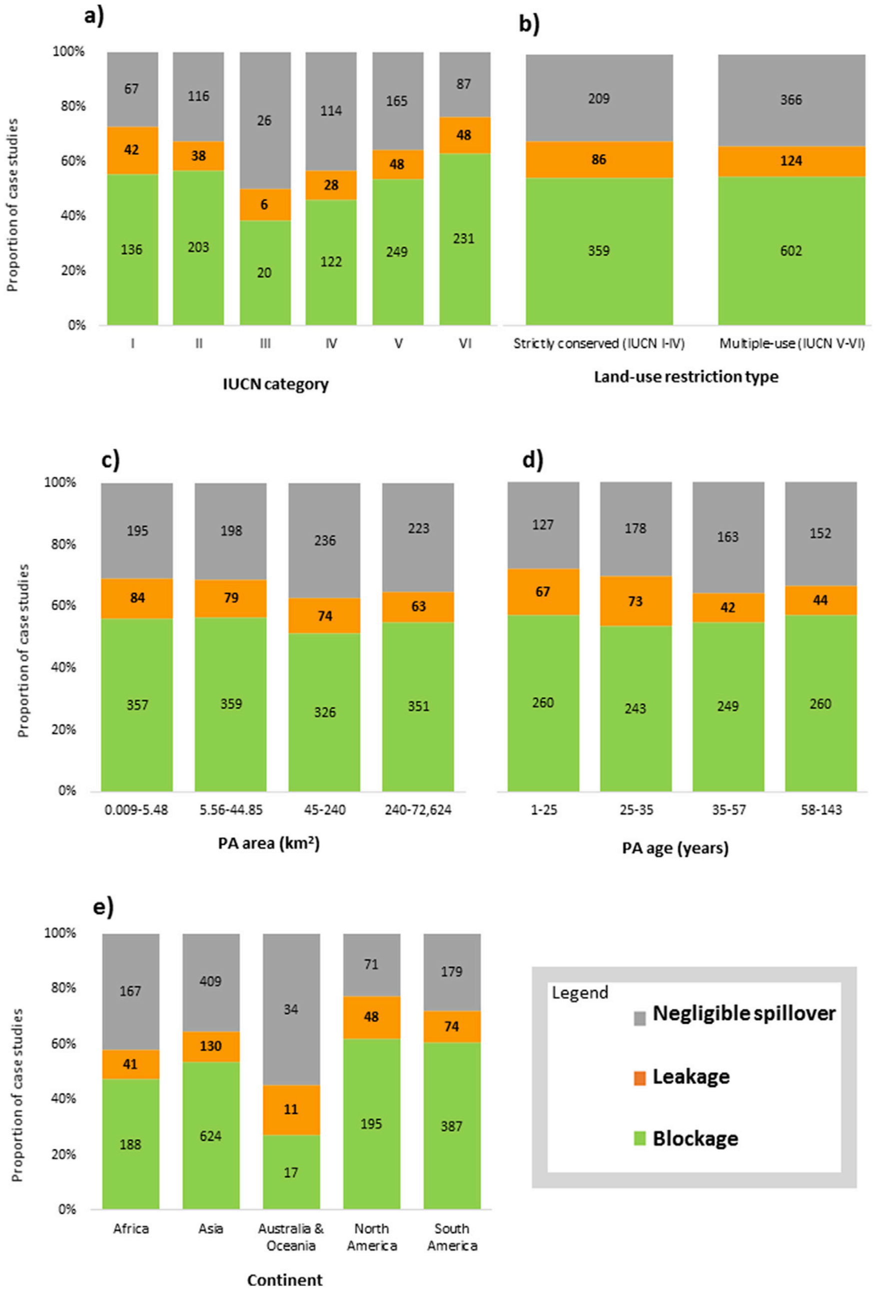

Fig. 5. Proportional distribution (and number per category) of spillover outcomes across PA-specific factors and continents. Results were broadly similar across: **a)** IUCN categories, **b)** land-use restriction types, whereby land use restriction is strict in PAs of IUCN categories I-IV and some degree of land use and resource extraction is allowed in PAs of IUCN categories V and VI, **c)** PA size quartiles**, d)** PA age quartiles, and **e)** continents. Each bar graph does not necessarily show the results for all 2,575 PAs (grand total assessed) because, for example, IUCN category was reported or found for only 1,746 of the PAs; PA size was reported for only 2,545 PAs; the year of establishment was reported or found for only 1,816 PAs.

4. Discussion

The conservation-science literature has recognised the possibility and implications of spillover effects, but few studies have attempted to quantify them. Our collated PA assessments, despite being the result of an exhaustive search, constituted <2% of the global PA network (UNEP-WCMC, 2016). Results from these 3,398 case studies affirmed the generally accepted view that most PAs reduce deforestation effectively within their borders relative to the wider landscapes in which they are embedded. Although we also identified some PAs that experienced higher deforestation within compared to without (the surrounding baseline), a more detailed case-by-case assessment is needed to draw firm conclusions about these possible

Table 3
Predictors included in the best-selected generalized linear models of leakage and blockage magnitudes. Color-shaded fields and the + and − symbols indicate the variables included in each model and direction of their effect.

	Leakage		Blockage	
	Complete cases of 15 variables	Complete cases of 13 variables	Complete cases of 15 variables	Complete cases of 13 variables
Number of observations	165	395	549	1 443
Percent of deviance explained	14.4	13.4	17.3	15.9
PA area				
PA age upon deforestation monitoring		**NA**		NA
Land-use restriction type		**NA**		NA
Population density	–	–	–	–
Population growth rate	+			+
Rural population growth rate	–		–	–
GDP				+
GDP growth rate	–		–	–
Ratio: % land area under agriculture to forest	+			+
Value of forest products removed in 1 year	+			
Value of forest products removed per hectare		+	+	
Carbon dioxide emissions				–
Percentage of energy supply from biofuels				
Annual conservation spending			–	–
Continent: Africa				
Continent: Asia			+	+
Continent: Australia & Oceania				+
Continent: North America				
Continent: South America				

“paper parks,” as prohibition of forest clearing might not have been the goal of protection (Appendix S4). Importantly, blockage occurred almost five times more frequently than leakage outcomes.

Because leakage comprises one pillar of the “unholy trinity” of challenges facing efforts to reduce deforestation (Van Oosterzee et al., 2012), we sought to characterise leakage-prone PAs by determining how features like IUCN category, size, and age since gazettal drove spillover direction towards either blockage or leakage. We were unable, however, to identify any consistent characteristics of highly leakage-susceptible PAs: the three outcomes—blockage, leakage, or negligible spillover—were distributed without apparent pattern across IUCN categories, land-use restriction types, size quartiles, and age quartiles (Fig. 5a–e). We also expected that one or more of these intrinsic PA characteristics might influence spillover magnitude. Instead, our modelling pointed to national-scale socioeconomic and political factors as being most relevant to spillover magnitude, for both leakage and blockage (Table 3). There is a logic to this outcome: displacement of human development pressure by land-use restriction is, after all, a socioeconomic phenomenon (Renwick et al., 2015), albeit one with ecological consequences. Attempts to predict spillover direction and magnitude for proposed or recently established PAs might therefore necessitate interdisciplinary partnerships between social scientists, economists, and natural scientists (Miteva et al., 2012) and assessment of potential predictors not considered in this study.

Our modelling results challenge the assumption of a trade-off between predictors associated with high leakage and low blockage values, or vice-versa (Table 3). None of the potential predictor variables demonstrated a relationship to leakage and to blockage indicative of a trade-off: PAs located in countries with higher population densities and higher rural population growth rates were associated with both lower leakage and lower blockage magnitudes (i.e., smaller-magnitude spillovers in both directions). PAs located in countries with higher population growth rates, higher ratios of land under agriculture compared to forest cover, and higher value-per-hectare forest products were all associated with both high leakage and blockage magnitudes (i.e., larger-magnitude spillovers in both directions). Case-specific analyses incorporating a socio-ecological framework (Ostrom, 2009) might be necessary to explain these complex dynamics.

Whilst significant uncertainty remains as to the magnitude and ubiquity of the leakage problem on global, national, and ecoregional scales, our results are promising and suggest that, at least in PAs characterized by subtropical and tropical moist forest cover, PA assessments should recognise the possibility not only of a problematic leakage outcome in the UAS, but also of a beneficial blockage outcome. Future studies could seek evidence to determine whether blockages are indeed generated by nearby protection changing perceptions of the development potential of the UAS, perhaps influenced by ecotourism (Herrera Garcia, 2015). There is clearly a need for more case studies of the non-target impacts of land-use restrictions associated with PA creation, and future spillover analyses should seek to calculate the most meaningful baseline possible. Our results were mostly extracted from deforestation data reported by one study (Spracklen et al., 2015) for which the wider landscape samples were arbitrarily chosen (Fig. 2b) and for which only PAs characterized by subtropical or tropical moist forest cover were assessed. Because the samples of the wider landscape (Fig. 2) did not necessarily experience the same likelihood of being deforested as the associated PA or the UAS experienced, and because the PA and UAS may have been less suitable for clearing due to slope, elevation, or some other factor (Joppa and Pfaff, 2009), it is possible that our impact calculations overstate the beneficial effect of protection and/or of nearby protection. Baseline estimates can be improved using statistical tools like propensity score matching (Andam et al., 2008; Chomitz, 2002; Joppa and Pfaff, 2010) and panel regression techniques (Jones and Lewis, 2015). Although our systematic literature search did identify a few studies that implemented propensity score matching methods, these studies did not meet the data-extraction criteria.

A baseline deforestation rate could also be derived temporally (Oliveira et al., 2007) rather than from samples of the wider landscape during the same period, although this involves potentially problematic assumptions due to the often non-linear dynamics of land-use change pressures over time. In the case of a temporally derived baseline, the deforestation rates in the PA and in the UAS prior to PA gazettal constitute the counterfactual expected deforestation rates. Comparison of temporally and spatially derived baselines was not possible in this review because the identified records did not report deforestation data for the period before PAs were created. A difference-in-difference baseline-estimation method (see Fig. 3 in Pattanayak, 2009), combining both the spatial and temporal means of informing the baseline estimate, offers a potentially useful way in which future spillover analyses might calibrate baselines. Further, we recommend that spillover monitoring begin when a PA is established and continue over longer durations than those typically reported in this review, as there is likely to be a time lag between protected status designation and the displacement of land-use pressures. Our sample did not provide enough variation in duration of deforestation monitoring to model its importance; all the case studies extracted in this review had 12 or fewer years of deforestation monitoring. Longer-term studies, beginning the year of PA establishment, would be ideal for clarifying the spatio-temporal displacement of land-use pressures.

In this review we quantified PA impacts using the metric of area deforested over time, but we acknowledge that human land use is more complex than the binary presence or absence of forest clearing. Furthermore, non-forest environments are also home to much of the world's biodiversity, and these habitats are also being degraded and destroyed due to the expansion and intensification of human land uses (Bond and Parr, 2010). Spillover analyses, and indeed PA assessments that reflect their net impact on the entire landscape, from a variety of perspectives and values, should reach beyond forested areas. Ideally, they should seek to capture the graded intensities of anthropogenic landscape modification using spatially explicit land-use and land-use-change (LULC) data, which is increasingly available from government initiatives such as Europe's Coordination of Information on the Environment (CORINE) and the United States Geological Survey. In addition to land-use restrictions by public PAs, protection associated with conservation covenants on private lands, with indigenous land governance, and with community managed forests, could also generate measurable spillover effects that may warrant quantification, collation, and meta-analysis.

Following identification of spillover effects around areas of land-use restriction, it is essential to then determine their biological and ecological consequences. As noted by Miteva et al. (2012), the degree to which measures of PA impacts on land-cover dynamics are biologically and ecologically useful is determined by the degree to which land cover is a good proxy for species richness and ecosystem function. Spatially explicit biodiversity data, like that provided by the IUCN for red-listed species' ranges (http://www.iucnredlist.org), could be employed to this end. The increasingly common practice of lodging such data in open-access repositories online (Ondei et al., 2018) will facilitate meta-analyses of the impacts of land-use restrictions on threatened species and habitats. Future studies may assess the relative strength of the mechanisms through which human land uses outside of PAs influence ecological function within them in terms of the effective size of the ecosystem being protected in part by the PA, flows of ecological process zones, crucial habitats, and exposure to humans at reserve edges (Hansen and DeFries, 2007).

Finally, this review focussed on local spillovers, sometimes called 'neighborhood spillovers', which although poorly understood are being addressed with increasing frequency in the conservation-science literature. Although difficult, future studies should also look beyond the problem of local leakage to track cross-biome and transnational leakages (Gan and McCarl, 2007; Meyfroidt et al., 2013). Cross-biome leakage might have occurred, for example, when sugarcane expansion in the state of Sao Paolo, Brazil affected land use in distant and disjunct Amazonas, due to the consequent displacement of cattle ranching (De Sá et al., 2013). Indeed, Dou et al. (2018) found evidence that reduced deforestation in the Amazon forest has come at the cost of increased deforestation in the Cerrado savanna. In this context, implementation of a tele-coupling framework (Bruckner et al., 2015) might be necessary to track non-local leakages and thus, ultimately, to balance trade-offs between development and conservation goals on a global scale.

5. Conclusion

Although the published global-ecology literature mostly discusses terrestrial spillovers in terms of the potential for leakages, we identified blockage as the most common outcome. It is promising that deforestation leakages were the least frequent consequence for the 2,575 PAs whose non-target impacts were assessed in this review. Nonetheless, PA spillover analyses ought to be done routinely given how critical they are for landscape-scale PA impact assessments and for monitoring the consequences of opportunistic, rather than systematic, protection of lands (Renwick et al., 2015). The goal of expansion of the global PA network is codified into international law by means of Target 11 of the 2010 Convention on Biological Diversity, but simply adding more hectares under protected designation does not necessarily translate to positive impacts for habitat and biodiversity, if they act to displace threats to areas where biodiversity is more highly vulnerable and irreplaceable. This should be a key focus of research as new regions are gazetted for conservation.

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Appendix A. Supplementary data

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